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Macroplastic accumulation in roadside ditches of New York State's Finger Lakes region (USA) across land uses and the COVID-19 pandemic

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ABSTRACT

Macroplastics are a ubiquitous and growing environmental contaminant with impacts in both marine and terrestrial systems. Marine sampling has dominated research in this field, despite the terrestrial origins of most plastic debris. Due to the high surface water connectivity facilitated by roadside ditches, these landscape features provide a unique sampling location linking terrestrial and surface water systems. We collected and analyzed macroplastic accumulation by number of pieces, mass, and polymer type in roadside ditches across four land uses, before and during the COVID-19 pandemic in the Finger Lakes Region of New York State. Commercial land use plastic accumulation rate was highest, while forested land use accumulation rates were lowest on a piece basis. Pre-COVID-19 piece accumulation rates were significantly higher than COVID-19 piece accumulation rates across all land uses. Mass accumulation rates followed similar patterns observed in piece accumulation, but the patterns were not always statistically significant. Plastic type 4 (i.e. thin plastic films), especially plastic bags and wrappers, was the most frequently collected type of macroplastic by piece across all land uses within the 1–7 Resin Identification Codes. By mass, the data were distributed less consistently across land uses. Cigarette filters, containing the polymer cellulose acetate, were the most frequently found roadside plastic, but are not within the 1–7 classification system. Our results suggest that policies in place limiting plastic bag usage could substantially reduce roadside plastics but other plastics, such as food wrappers and other single use plastic films, which comprised a large proportion of the plastic debris collected, should also be regulated to further decrease macroplastic pollution.

1. Introduction

Plastic pollution is a global issue that shows little evidence of dissipating in the near future (Thompson et al., 2009). According to PlasticsEurope, a pan-European plastics trade association, global plastic production increased from an average of 2 million metric tons in the 1950s to nearly 360 million metric tons in 2017, a 200-fold increase, with only 9 % of total disposed plastic successfully recycled since 1950 (Ritchie, 2018; Shen et al., 2020). Since 1991, there has been international momentum to reduce the prevalence of single-use plastic bags, with countries around the world instituting usage restrictions (Bezerra et al., 2021; Clayton et al., 2020; Adam et al., 2020; Schnurr et al., 2018; Xanthos and Walker, 2017). Beginning with California in 2014, eight U. S. states have now banned single-use plastic bags (Schultz and Tyrrell, 2020), with New York State following in 2020 (Article 27 Title 28 Bag Waste Reduction, 2020). Plastic bags were identified as plastic type 4 according to the Resin Identification Code (RIC) created by the Plastic Industry Association (1988).

The negative effects of plastic pollution abound in the environment. For example, macroplastics kill over 1 million marine animals each year due to plastic related ingestion and entanglement (Efferth, 2017). Holland et al. (2016) found that 11.1 % of sampled freshwater birds across Canada had ingested plastic debris. Plastic decomposition is generally slow and varies between air and water environments due to

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differential photodegradation of plastic polymers (Biber et al., 2019; Tang et al., 2019) with some estimates suggesting 400 years for plastics to decompose in marine systems, prolonging detrimental impacts (Barnes et al., 2009). Macroplastics decompose first into smaller plastics (i.e. microplastics, nanoplastics) and then into their constituent plastic polymers. Plastic particles provide adsorption surfaces for persistent organic pollutants (Koelmans et al., 2016; Yu et al., 2019), while their constitute polymers can be toxic even after microbial degradation (Kale et al., 2015). Henseler et al. (2019) found macroplastic pollution from improper disposal of household trash entered agricultural soil, where it fragmented and was subsequently transported through runoff, airborne distribution, drainage systems, and erosion. Henseler et al. (2019) also found microplastics leached into the soil, where they reduced soil productivity, diminished agricultural crop yields, and were occasionally ingested with food crops. Ingested plastics and their associated contaminants can bioaccumulate (Teuten et al., 2007), leading to human gastrointestinal tract inflammation, internal tissue abrasion, and liver damage in some cases (Revel et al., 2018).

Studies have begun to assess the significance and scale of plastic pollution (e.g. Cable et al., 2017; Ryan et al., 2019; Sorensen and Jovanović, 2021; Watkins et al., 2019), with a primary focus on microplastics (Horton et al., 2017), i.e., plastics smaller than 0.5 cm in diameter. However, Blettler et al. (2018) reveal a lack of macroplastic research, specifically in terrestrial systems (Alimi et al., 2018). Macroplastic pollution has predominantly been documented in oceans (e.g., Barnes et al., 2009; Lebreton et al., 2019; Ostle et al., 2019). An oceanographic model of floating debris dispersal estimated over 5.25 trillion plastic particles weighing more than 268,940 metric tons on ocean surfaces alone, which excluded plastics that sank or were deposited on shores (Eriksen et al., 2014). It was estimated that between 19 and 23 million metric tons of plastic pollution from 2016 entered aquatic ecosystems from global plastic production, and that the correlated emissions will continue to increase (Borrelle et al., 2020). Most marine plastic pollution originates on land, indicating that terrestrial macroplastic monitoring and mitigation is critical to develop impactful solutions for overcoming both terrestrial and marine macroplastic pollution issues (Hurley et al., 2020; Jambeck et al., 2015). A more holistic understanding of terrestrial macroplastic dynamics is a critical knowledge gap in the plastic literature (Alimi et al., 2018; Horton et al., 2017; Malizia and Monmany-Garzia, 2019).

Roadside ditches are designed to minimize flooding risk to roads by rapidly transporting runoff from upland areas to downstream water bodies. Buchanan et al. (2013a) found that roadside ditches drained 27 % of a small watershed characteristic of Central New York, located approximately 40 km from our study sites. Roadside ditches play a critical role in the transport of nonpoint source pollution (Buchanan et al., 2013b; Falbo et al., 2013), which likely includes macroplastics. Liro et al. (2020) modeled macroplastic input, transport, storage, remobilization and output from riverine systems. Roadside ditches connect upland terrestrial systems to rivers, providing a macroplastic input sampling point into headwater streams and rivers. We chose to sample in ditches because they probably disproportionately accumulate plastics relative to the surrounding landscape and act as pollution conduits to downstream water bodies.

Concerning the ongoing global COVID-19 pandemic, Silva et al. (2021) addressed concerns over increased single-use plastic consumption and disposal, especially personal protective gear for increased safety measures, as a result of COVID-19. The introduction of personal safety measures encouraged by scientists and professionals, specifically through the mass usage of single use face masks, was estimated to increase the amount of waste by 95 % (Silva et al., 2021). Prata et al. (2020) hypothesized that with the ongoing COVID-19 pandemic, there is an associated increase in single-use plastic use resulting from restaurants and other hospitality sectors shifting to single-use plastics for public health reasons. This led to the estimate that plastic packaging may have increased by 14 % in the United States alone (Prata et al., 2020). These estimates posed the question of how quarantine restrictions altered terrestrial environmental plastic accumulation due to modifications in human behaviors.

We assessed the impact of differing land uses on number, mass, and type of macroplastics accumulated in roadside ditchesOur data span periods before and during quarantine restrictions implemented as a result of the COVID-19 pandemic. Our objectives are to (1) identify locations characterized by high macroplastic accumulation rates and (2) identify differences in plastic pollution one year before and one year after the onset of the COVID-19 pandemic. Our goal is to provide information that directs effective plastic pollution mitigation efforts. We also considered correlations between traffic volume (vehicles/hr) and plastic accumulation.

2. Methods

2.1. Study sites

We selected twelve roadside ditches in Tompkins County, NY across four land uses. Tompkins County resides in the Finger Lakes region of NY with a cool, wet climate and a yearly rainfall of 21–31 inches (Whitesell, 2005). Sites were located in the Six Mile Creek, Cascadilla Creek, and Fall Creek watersheds (Fig. 1, Table 1).

We identified four dominant land uses (agricultural, commercial, forested, and residential) from the Tompkins County Natural Resources Inventory maps (Tompkins Co, 2020). The agriculture land use included cropland, cattle, horse farms, fisheries, horticulture, inactive fields, orchards, pastures, tree farms, vineyards, and other farm types. Commercial land use included malls/shopping centers, commercial storage, retail and office buildings. Forest/Shrub land use includes deciduous and coniferous forests, mixed forests, forest plantations, brush and grasslands. Residential land use includes low to high density areas and manufactured home parks (Tompkins Co., 1999). We selected three roadside ditches in each land use. We selected ditches approximately 2 m wide, and uninterrupted for at least 60 m in length. We measured traffic (cars/hr) during sample collection at each site (Table S3). The land defined as "Other" in Fig. 1 includes industrial, public/private institutional, water/wetlands, outdoor recreation, transportation, disturbed/barren lands, and abandoned buildings. Ditches in these land uses were not considered in this study.

2.2. Site hydraulics

We recorded ditch characteristics including channel slope, average depth, average width, and vegetation composition (Table S1). We measured the depth and width at each site with a survey rod and a scope at 10 m, 30 m, and 50 m, then calculated the average for each site. We measured average channel slope between each surveyed section. Manning's roughness coefficients were estimated from channel materials and vegetation composition (Chow, 1959).

2.3. Macroplastics sample collection

Collected macroplastics spanned the categories of mesoplastic (between 4.75 mm and 200 mm in diameter) and macroplastic (greater than 200 mm in diameter) identified by Eriksen et al. (2014), i.e., any plastic particle we could visually identify was considered a macroplastic. Macroplastics were manually collected at all sites every two to three weeks from June 2019 to December 2019 (8 sampling rounds; pre-COVID-19) and August 2020 to November 2020 (3 sampling rounds; COVID-19) (Table S2). All litter was removed from each site prior to the first sampling campaign (June 2019) to establish a zero-litter baseline from which to compare plastic accumulation between subsequent collections. Accumulation rates were calculated for pieces (pieces/day) and mass (g/day) between collections after the initial baseline establishment. Due to COVID-19 restrictions, time since previous sampling was

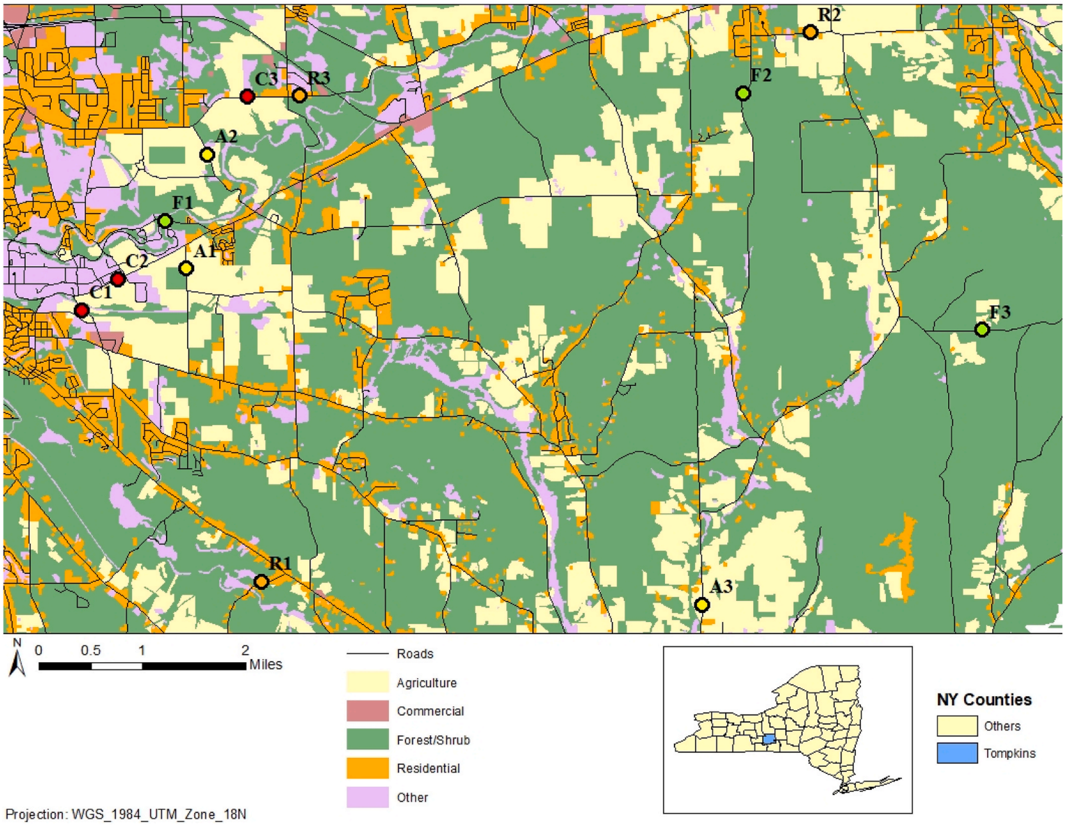

Fig. 1. Site map displaying land use types designated by Tompkins County Land Use and Land Cover Mapping Project (Tompkins Co, 2020) and major roads. Sampling locations are denoted with colored circles.

longest (9 months) for the first COVID-19 sample (August 2019) and the number of collections in the COVID-19 data subset were less than the pre-COVID-19 number of collections due to quarantine restrictions during the COVID-19 pandemic. There was no initial purging of plastics at the beginning of the COVID-19 data collection. We recorded the number of plastics, plastic mass, plastic dimensions (length, width, and thickness), and plastic type. Samples were washed with tap water to remove sediment and oven dried at 50 °C for at least 24 h before weighing.

We collected plastics along 60 m of road ditch at each site over a width spanning 1 m on both sides of the ditch center (120 m^2). This sampling scheme extended beyond some ditch widths. Only plastic samples were analyzed, although all types of litter were removed.

We categorized plastic type by the Resin Identification Code (RIC) set forth by the Plastic Industry Association (1988) by visually and textually matching samples to RIC categories. The numerical classes (1–7) indicate plastic recyclability, defined as the ability of a material to reacquire its previous properties after the recycling process. The included categories are (1) polyethylene terephthalate (e.g., water bottles, vegetable oil containers), (2) high density polyethylene (e.g., household cleaner, shampoo bottles), (3) polyvinyl chloride (e.g., vinyl pipes, thick bottles), (4) low density polyethylene (e.g., plastic bags, frozen food bags, wrappers), (5) polypropylene (e.g., caps, straws), (6) polystyrene (e.g., styrofoam; takeout containers), and (7) polycarbonate (e.g., large containers, mixed plastic materials, most other plastics).

Specific descriptions were attached to each plastic piece to allow interpretation of the variety of plastics within the same RIC plastic type. These descriptions were clustered into categories outlined by NOAA marine Debris monitoring protocol (Opfer et al., 2012). Plastic fragments with plastic bag-like characteristics were commonly found in roadside ditches, were classified as "bag pieces" if it was clear they came from plastic bags. Otherwise, they were classified as "thin film plastics". Many small, film-like plastics likely were from plastic bags and other films that had been mowed and shredded. We also collected cigarette filters, categorized as cellulose acetate, the polymer comprising most filters, which is very difficult to recycle (Mohajerani et al., 2016). Cellulose acetate data were synthesized separately due to exclusion of this polymer from the RIC categorization system.

2.4. Statistical analyses

Statistical analysis was performed with the R scripting language (R Core Team, 2013). A Shapiro-Wilk test was used to determine that the data were not normally distributed.

Using the Wilcoxon Rank Sum test and verified using a Kolmogorov-Smirnov (K–S) test (Table 2, Figure S2, Figure S3), we assessed statistical differences in plastic accumulations between land uses in the same sampling period and between pre-COVID-19 and COVID-19 samples of the same land use. We considered p-values < 0.05 statistically significant and denoted p-values between 0.05 and 0.01 with (*), p-values between 0.01 and 0.001 with (**), and p-values less than 0.001 with (***).

To assess the influence of land use on cellulose acetate accumulation, a Kruskal-Wallis Rank Sum test was used on cellulose acetate pieces grouped by land use. The statistical significance of the p-value was ranked based on the same standards for the Wilcoxon Rank Sum test.

To analyze the distribution of plastic pieces by type, we used a Fisher Exact Test to determine independence of the distributions of plastic type compared within a land use across time period (e.g., agricultural pre-COVID-19 samples versus agricultural COVID-19 samples) and within a time period across land use (e.g., pre-COVID-19 samples compared between agricultural, commercial, forested, and residential land uses). The null hypothesis assumed independence between plastic type and the tested variable (time period, land use). The null hypothesis was rejected

Table 1
Sampling locations and characteristics.

ID	Surrounding Land Use	Coordinates	Road and Municipality	Description
R1	Residential	42°24′18.2″N 76°25′48.8″W	German Cross Rd, Ithaca, NY	Rip-rap base material with unmowed vegetation. Intermittent flow.
R2	Residential	42°28′55.7″N 76°19′54.5″W	Ferguson Rd, Dryden, NY	Vegetated with very dense vegetation on side slopes. Earthen base. Stagnant, ponded water in summer; dry fall through winter.
R3	Residential	42°28′19.6″N 76°25′44.7″W	Hanshaw Rd, Ithaca, NY	Stagnant, ponded water. Vegetated side slopes with mud bottom.
C1	Commercial	42°26′30.0″N 76°28′08.0″W	Pine Tree Road, Ithaca, NY	90 % vegetated, 10 % “rip-rap”
C2	Commercial	42°26′44.9″N 76°27′45.1″W	Cornell Univ. Orchards, Ithaca, NY	30 cm tall grass, one mowed side slope, one unmowed side slope
C3	Commercial	42°28′18.9″N 76°26′14.9″W	Hanshaw Rd, Ithaca, NY	Very deep with bare earth bottom. Flowing water June–August; very wet March–May.
A1	Agricultural	42°26′40.3″N 76°26′56.0″W	Game Farm Rd, Ithaca, NY	Stagnant, ponded water. Earthen, vegetated bottom
A2	Agricultural	42°27′54.4″N 76°26′46.0″W	Freese Rd, Ithaca, NY	Earthen, vegetated bottom and sides. Stagnant, ponded water present.
A3	Agricultural	42°24′06.6″N 76°21′01.2″W	Midline Rd, Slaterville Springs, NY	Deep ditch with sandy, earthen bottom. Unmowed grass and rock lined side slopes.
F1	Forested	42°27′15.0″N 76°27′03.8″W	Forest Home Dr, Ithaca, NY	Shallow earthen ditch with patchy short vegetation.
F2	Forested	42°28′24.9″N 76°20′38.4″W	Yellow Barn Rd, Dryden, NY	Unmowed, earthen ditch. No flowing water.
F3	Forested	42°26′27.0″N 76°17′56″W	Star Stanton Rd, Freeville, NY	Stagnant, ponded water at bottom of the gradient. Vegetated side slopes; rip-rap base.

Table 2
Wilcox rank sum results for Pre-COVID-19 compared to COVID-19 samples for piece and mass accumulation within the same land use.

Land Use	Piece accumulation p-value	Mass accumulation p-value
Agricultural	0.00148**	0.517
Residential	0.000455***	0.536
Commercial	0.00585*	0.0658
Forested	0.0184*	0.0282*

at $p < 0.05$ in favor of the alternative hypothesis displaying dependence between the two tested variables.

3. Results

Over the course of the study, we collected 473 plastic pieces categorized into RIC types 1–7. We collected an average of 11.8 pieces per round per site, with a standard deviation of 13.8 pieces per site per collection. One site (Site F3) consistently collected no plastic pieces. Several sites had one piece per collection (Site A3, F1, F2, F3, R1, R2, R3), while the maximum pieces collected at one site was 70 plastic pieces per collection (Site C2). The total mass collected was 2323.6 g, with an average of 23.95 g of plastic per collection, and a standard deviation of 61.92 g across all sites and rounds. The minimum mass collected was 0.02 g (Site R3) and the maximum mass was 566.02 g (Site C3).

The total amount of cellulose acetate collected was 674 individual pieces and the total mass collected was 162.1 g. The average pieces collected per collection per site was 10.9 pieces, with a standard deviation of 11.4 pieces. The minimum collected at a site was 1 piece (Site A1, A2, A3, C3, F1, F2), and the maximum collection value at a site was 48 pieces (Site C1). The minimum mass of cellulose acetate collected at a site was 0.13 g (Site F1), and the maximum mass of cellulose acetate collected at a site was 12.45 g (Site C3).

3.1. Land use

We found statistically significant differences (p-value < 0.05) in plastic piece accumulation in the pre-COVID-19 period between agricultural and commercial land uses (p-value = 0.004**), commercial and forested land uses (p-value = 0.002**), commercial and residential land uses (p-value = 0.018*), and forested and residential land uses (p-value = 0.037*). By contrast, COVID-19 plastic piece accumulations were significantly different only between commercial and residential (p-value = 0.008**) and commercial and forested (p-value = 0.026*) land uses (Fig. 2A).

Pre-COVID-19 plastic mass accumulations were significantly different between agricultural and commercial land uses (p-value = 0.014*) and between commercial and forested land uses (p-value = 0.035*). COVID-19 plastic mass accumulations were significantly different between commercial and forested (p-value = 0.016*), agricultural and forested (p-value = 0.032*), and forested and residential (p-value = 0.023*) land uses (Fig. 2B).

Total plastic piece accumulation summed across land uses during pre-COVID-19 sampling was significantly higher than during COVID-19 (p-value < 0.0001***). For mass accumulation between collection periods, a p-value of 0.0133* was calculated, indicating that the rate of

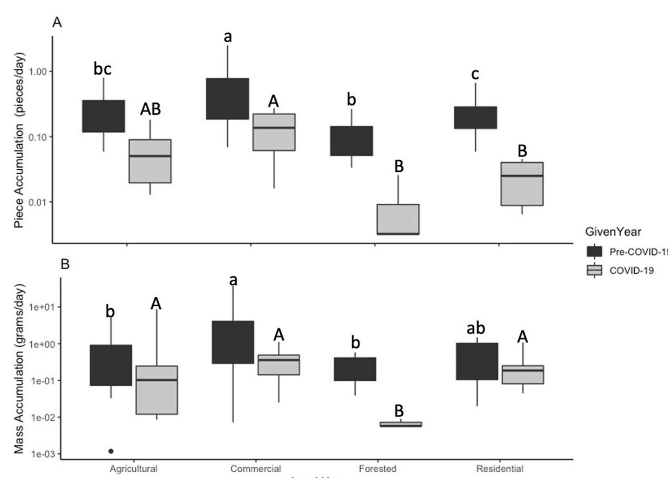

Fig. 2. Piece (pieces/day) (A-top) and mass (g/day) (B-bottom) accumulations across land uses and pre-COVID-19 (light bars) and COVID-19 (dark bars) years. Different letters above the bars indicate significant differences between land uses; lower case letters (a,b,c) reference pre-COVID-19 data, while capital letters (A,B) reference COVID-19 data. Statistical difference within a collection period is noted by different alphabetical letters. Note: all pre-COVID-19 piece accumulations (A-top) were higher than COVID-19 piece accumulations for each land use; pre-COVID-19 mass accumulations (B-bottom) were only significantly higher for forest land uses.

COVID-19 piece accumulation was less than the pre-COVID-19 mass accumulation rate when summed across all land uses.

By land use, all piece accumulations were statistically higher during the pre-COVID-19 period than the COVID-19 period (Table 2). For mass accumulations, only forested pre-COVID-19 samples were statistically higher than COVID-19 samples (Table 2), though there is a non-statistically significant trend of lower mass accumulations across all land uses between in the pre-COVID-19 period when compared to the COVID-19 period (Fig. 2B).

3.2. Plastic type

Overall (pre-COVID-19 and COVID-19), plastic type 4 (e.g. plastic bags, lighter weight plastic films, wrappers) was the most frequently found by piece in roadside ditches. Type 4 plastics comprised 55.7 % of plastics at agricultural sites, 42.8 % at commercial sites, 51.8 % at forested sites, and 35 % at residential sites across both time periods. When considering percent by mass, types of plastic with the most mass by land use were more variable. We found agricultural sites had 34.7 % type 5, commercial sites had 54.6 % type 5, forested sites had 78.9 % type 4, and residential sites had 33.8 % type 1 as their most prevalent plastic type by mass.

When comparing plastic type distribution by piece between pre-COVID-19 and COVID-19 samples for each land use, we found that distributions at agriculture (p-value = 0.93) and commercial (p-value = 0.061) land uses did not statistically shift with regards to plastic type between pre-COVID-19 and COVID-19 time periods. However forested (p-value = 0.0093**) and residential (p-value = 0.0093**) land uses were defined by plastic type distributions that statistically differed between these two periods. Distributions at forested land uses shifted to favor plastic type 6 in the COVID-19 period over types 1 and 5; while residential land use plastic type distributions shifted to favor plastic types 1, 3 and 6 over types 2 and 4 (Fig. 3A).

When comparing plastic type distributions by piece between land uses of the same time period, we found that land use significantly predicted plastic type distribution in the pre-COVID-19 period (p-value = 0.038*) while land use did not significantly predict plastic type distribution in the COVID-19 period (p-value = 0.126) (Fig. 3A).

Plastic type distributions by mass were not statistically compared due to the dominance of singularly heavy individual pieces in the distributions. Instead, mass by type distributions are shared to qualitatively compare the distributions across time period, land use, and between piece and mass data. Plastic type 4, though dominant in the piece distributions, are lightweight and tend to have diminished importance in the mass by type distributions in comparison. The one anomaly to this trend was in the forested sites, that had similar type distributions across piece and mass analysis, but also had fewer plastic particles (18 pre-COVID-19 plastics) than agricultural (83 pre-COVID-19 plastics), commercial (192 pre-COVID-19 plastics), or residential sites (68 pre-COVID-19 plastics).

We found a total of 674 cigarette filters (cellulose acetate; the most numerous of any polymer found by piece), 489 in the pre-COVID-19 period and 185 during the COVID-19 period. By piece, we found 67.95 % of total collected cellulose acetate at commercial land uses, 22.00 % at agricultural, 8.01 % at residential, and 1.04 % at forested land uses across both time periods. By mass, they were found in 69.08 % commercial, 22.93 % agricultural, 7.32 % residential, and 0.67 % forested sites across both time periods.

Fairly consistently, the amount of cellulose acetate found for at each land use was higher during the Pre-COVID-19 collection period. In comparison between land uses (Fig. 4), commercial had the highest amount of cellulose acetate collected by piece. There was little variance for residential and forested land uses, as seen in by the whiskers on the plot.

We found statistically significant differences (p-value < 0.05) in plastic piece accumulation in the pre-COVID-19 period between agricultural and commercial land uses (p-value = 0.037*), agricultural and forested land uses (p-value = 0.014*), agricultural and residential (p-value = 0.044*), commercial and forested land uses (p-value = 0.012*), commercial and residential land uses (p-value = 0.013*), and forested and residential land uses (p-value = 0.026*) (Fig. 4). By contrast, COVID-19 plastic piece accumulations were not found to be significantly different across land uses.

We also found statistically significant differences in plastic piece accumulation within land uses between the pre-COVID-19 period and the COVID-19 period in residential (p-value = 0.0446*), commercial (p-

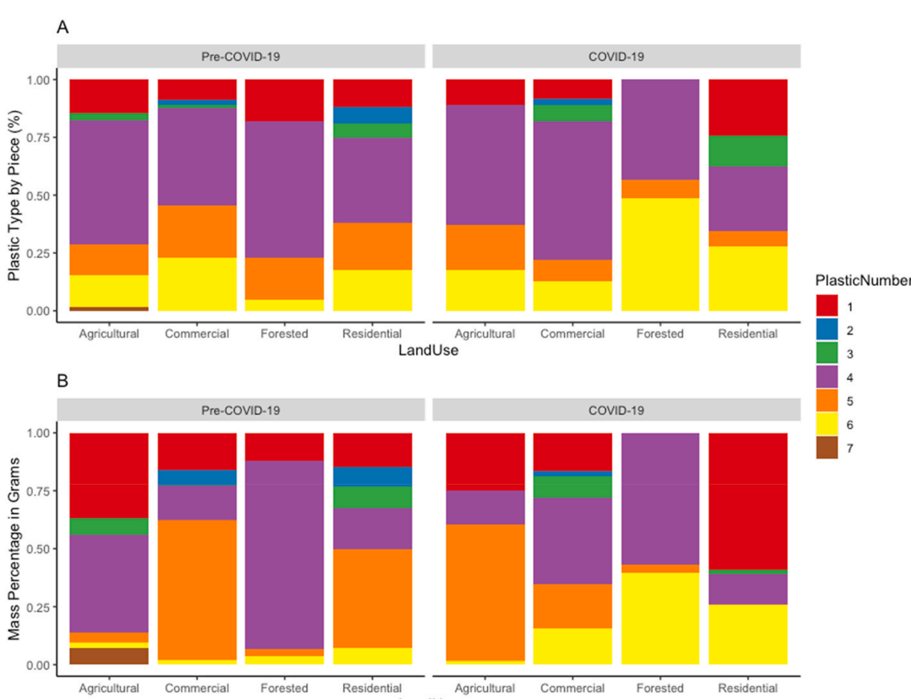

Fig. 3. Percent plastic type by peice (A) and mass (B) grouped by land use to make up total amount of plastic amount and mass. Plastic numbers are associated with the following plastics uses: (1) e.g., water bottles, vegetable oil containers, (2) e.g., household cleaner, shampoo bottles, (3) e.g., vinyl pipes, thick bottles, (4) e.g., plastic bags, frozen food bags, wrappers, (5) e.g., caps, straws, (6) e.g., styrofoam; takeout containers, and (7) e.g., large containers, mixed plastic materials, most other plastics.

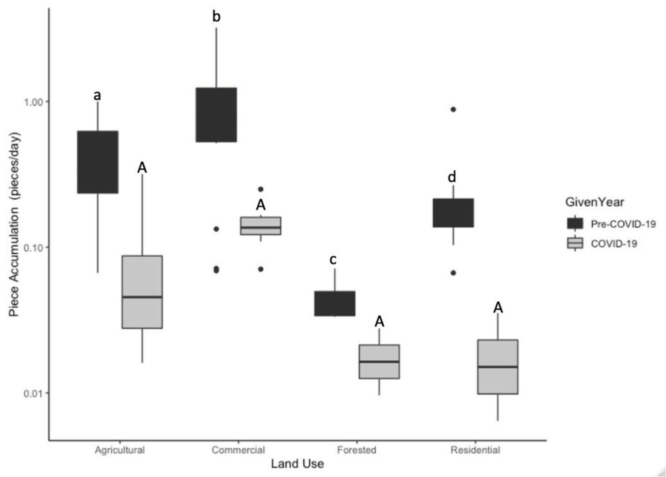

Fig. 4. Cellulose acetate piece accumulation (pieces/day) across land uses and pre-COVID-19 (light bars) and COVID-19 (dark bars) years. Different letters above the bars indicate significant differences between land uses; lower case letters (a,b,c) reference pre-COVID-19 data, while capital letters (A,B, C) reference COVID-19 data.

value = 0.00706**) and agricultural (p-value = 0.00319**) land uses. No difference was found between collection periods for the forested site. Cellulose acetate was found at 11 of the 12 sites, with site F3 having no cellulose acetate present.

Anecdotally we found a slight increase in personal protective medical equipment (i.e., masks, gloves) in the COVID-19 period. Two gloves were collected pre-COVID-19 (Site A3) while three were collected during the COVID-19 period (Site C1). No surgical face masks were collected pre-COVID-19, while two were collected during COVID-19 (one at Site A1 and one at Site C2). Across both periods, the most common individual plastics found were wrappers, Styrofoam, and bags. Wrappers and bags are both classified as plastic type 4, while Styrofoam is plastic type 6 (Fig. 5). Bag pieces were the fourth most frequently found plastic by description. Combined across land uses, there were no statistical differences in the plastic classifications used by NOAA marine classification system between pre-COVID-19 and COVID-19 periods (p-value = 0.1550) (Fig. 5).

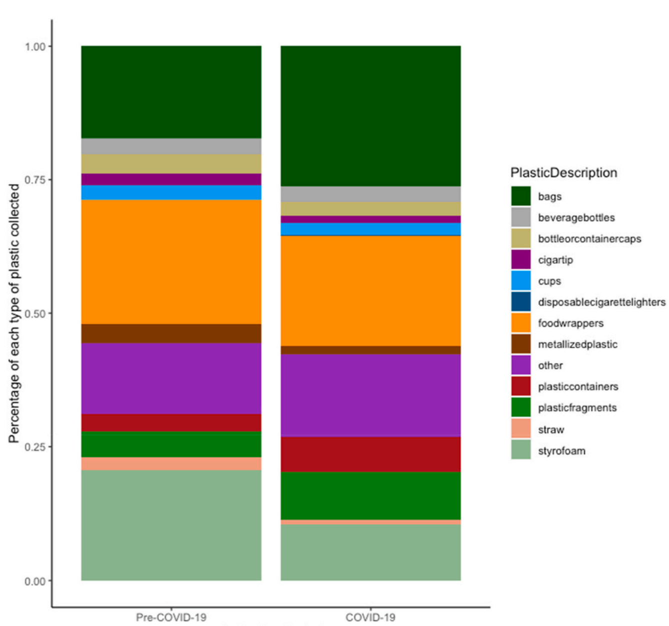

Fig. 5. Plastic piece classified by NOAA Shoreline Survey Field Guide classifications (Opfer et al., 2012) between pre-COVID-19 and COVID-10 periods.

4. Discussion

We found that commercial sites collected more plastic pieces than other land uses, but that by mass, there were fewer statistical differences in accumulation rates between land uses (Fig. 2). This was true for both pre-COVID-19 and COVID-19 collections. Pre-COVID-19 and COVID-19 data comparisons within the same land use showed that piece accumulation rates changed significantly between the two collection periods (Fig. 2). However, mass accumulations did not show statistically significant differences between collection periods or land uses. We found that type 4 plastics were the most prevalent plastics found by piece across all land uses, with some plastic type distribution differences between pre-COVID-19 and COVID-19 periods, and within periods across land uses (Fig. 2). These patterns were also observed for correlations between plastic accumulation and traffic (cars/hr), although the data were more variable. This result agrees broadly with other studies reporting that polyethylene film was the most common form of microplastic at an agricultural farmland (Piehl et al., 2018) and even within soil collected at a remote island conservation area in Norway (Cyvin et al., 2021).

Our analysis of pre-COVID-19 and COVID-19 samples showed the pandemic significantly reduced plastic accumulation in roadside ditches. Due to high travel restrictions and modified daily traffic flow during COVID-19, this result is expected. Silva et al. (2021) found that the COVID-19 pandemic dramatically increased single use plastics, especially in medical scenarios. However, our data suggest that despite this increased usage of personal protective equipment, overall plastic litter was reduced as a result of COVID-19 restrictions on personal travel or other behavioral changes caused by the pandemic. However, as predicted by Silva et al. (2021), we observed an overall increase in single use medical equipment (i.e., surgical face masks and gloves). Both sites where personal protective equipment was found (commercial and agricultural) were also characterized by more traffic (Figure S1). This supports the conclusion that higher traffic increased pollution of all kinds.

Cellulose acetate (i.e., cigarette filters) showed a similarly significant trend in reduction form pre-COVID-19 to COVID-19 samples within the same land use as was observed in general plastic piece accumulation. The trends seen in general plastic piece accumulation across land uses of the same sampling period also holds for cellulose acetate, however there were greater differences between land uses for cellulose acetate with more significant differences.

Numerous reviews have identified a lack of primary research on terrestrial macroplastics (e.g., Alimi et al., 2018; Bucci et al., 2020, Horton et al., 2017; Malizia and Monmany-Garzia, 2019; Winton et al., 2020). Our study addresses understudied trends in terrestrial macroplastic accumulation, specifically in roadside ditches adjacent to agricultural, commercial, forested, and residential land uses. While this a growing—albeit still limited—literature examining the transport routes and dynamics of terrestrial macroplastics (e.g., Lechthaler et al., 2020; Liro et al., 2020), much of this research examines the flow of plastics along rivers (Emmerik et al., 2019; Lebreton et al., 2019; Vriend et al., 2020) and storage in floodplain sediments (e.g., Ebere et al., 2019). While fluvial transport is the primary mode of terrestrial microplastic movement,our results begin to answer questions around other ancillary sources of terrestrial and freshwater microplastic generation, an underexamined area of plastic research (Xu et al., 2020). Future studies are required to address accumulation beyond ditches in other landscape features. Protocols used in marine plastic sampling must be modified to fit terrestrial, non-flow through environments, however classification systems already in place for use in marine plastic pollution may be directly transferred into this understudied area. In beginning to address the distribution of terrestrial macroplastic pollution, our study has created a holistic view of influencers, impacts, and trends in terrestrial pollution.

4.1. Obstacles in describing terrestrial plastic accumulation

Tracking ditch management (e.g., mowing, scraping, cleaning) would enable broader conclusions from results and potential explanations for differences between mass and piece accumulations. We noted that some ditches were mowed periodically and others were scraped out (Table S1), which could have shredded or removed plastic from the sampling locations respectively. Shredded plastic pieces may be more easily transported due to their reduced size and mass. Heavy rainfall events may also have transported plastics into ditches or downstream. Plastic transport was especially likely between pre-COVID-19 and COVID-19 sampling. To understand the influence of particle transport, future studies might consider monitoring plastic accumulation before and after major storm events. This potential transport of particles makes it difficult to precisely measure and display the scope of harmful effects of plastic pollution, though trends can be observed, as discussed elsewhere (Henseler et al., 2019; Lechthaler et al., 2020; Liro et al., 2020).

Snow coverage limited the effective sampling period to non-winter months. Snow cover December through March prevented collection of all macroplastics, so we paused sampling until conditions changed. If sampling is to occur throughout winter months in snow dominated regions, researchers should consider transport of plastic by snow plows during that period.

4.2. Study implications

We found that plastic type 4, the plastic type associated with plastic bags and other films, was the most commonly found plastic piece in the RIC 1–7 categories in roadside ditches (Fig. 3A). In response to the recent onset of plastic bag bans worldwide, our study, in part, supports this movement due to the number of type 4 plastics collected. With stricter regulation of plastic bag and film production and consumption, plastic piece pollution may decrease in roadside ditches.

To input our data into the model produced by Liro et al. (2020), transport potential within ditches should be calculated and surrounding land uses considered. Road ditch characteristics such as slope, vegetation, and flow conditions observed in our study can be used to estimate risk of transport downstream, and connects terrestrial plastic pollution to surface water pollution. Then risk of accumulating versus transporting plastics would be more accurately predicted along with macroplastic loads to water bodies.

The amount of thin, film-like plastics that could be shredded parts of plastic bags reveals that plastic bag bans are a good first step to reduce prevalence of type 4 plastics, but the number of wrappers and other films suggests plastic pollution mitigation requires expanded policy initiatives targeting other type 4 plastics. The most prevalent plastic type by mass varied across land use. Because of the variance in data collected, it would be difficult to suggest policy decisions that would restrict plastic consumption on a basis of mass.

We recommend plastic pollution mitigation focus on commercial, higher trafficked, commercial sites to maximize plastic removal from the environment. We have outlined trends in plastic pollution across land uses and traffic patterns. Future studies should address how water flow will impact transport of macroplastics through roadside ditches.

5. Conclusions

Our analysis revealed the complexity of quantifying plastic pollution in the terrestrial environment. We found notable correlations between human activity and plastic pollution. We found that commercial sites (which had the most traffic) accumulated significantly more plastic pieces than other land-uses. However, plastic accumulation by mass showed less statistical differences across land uses. In addition, we found that COVID-19 significantly lowered plastic accumulation rates by piece as pre-COVID-19 counts were significantly higher than COVID-19 counts. Cellulose acetate was by far the most collected polymer by piece in the form of cigarette filters. Our support policy decisions based on plastic piece accumulation. Decision makers may want to prioritize commercial areas due to their propensity for plastic piece accumulation. We observed a positive correlation between traffic (cars/hr) and plastic piece accumulation rates. We found that type 4 plastics were the most collected plastic in every land use by piece. This reveals that plastic bag bans target the correct plastic type, although expansion of this policy to include other kinds of type 4 plastics such as packaging and food wrappers may aid in overall pollution reduction.

Declaration of competing interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

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Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.jenvman.2021.113524.

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